MADROÑO, Vol. 56, No. 2, pp. 89–98, 2009

# EFFECTS OF FIRE AND GROUNDWATER EXTRACTION ON ALKALI MEADOW HABITAT IN OWENS VALLEY, CALIFORNIA

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## ABSTRACT

Alkali meadow habitat—a groundwater dependent ecosystem—is rare in California, and its response to fire has not been documented. We sampled vegetation in this habitat across a pumping-induced depth-to-water (DTW) gradient immediately before, then eight weeks after, a summer wildfire. After the fire in the burned area, we documented vigorous re-growth of native perennial grasses in areas of shallow groundwater but no re-growth of shrubs. The dominant grass *Sporobolus airoides* flowered in the shallow end of the DTW gradient but never advanced beyond vegetative phenology (leaves) in the drawn-down end. DTW explained 77% of the grass cover variance in the post-fire burned area, and 87% and 94% of the grass cover variance in the pre-fire and post-fire unburned (control) areas, respectively. This suggests that post-fire re-growth was re-establishing a cover-DTW relationship already present before the fire. The principal short-term fire effect was the elimination of shrub cover due to apparent shrub mortality. Our study shows fire may be an effective management tool for regenerating alkali meadow in areas of shallow groundwater. However, in areas subject to long-term water table drawdowns we found negligible grass re-growth and increased vulnerability to erosion, suggesting fire may accelerate the process of type-conversion from meadow to xeric shrubland.

Key Words: *Distichlis spicata*, fire, groundwater-dependent, groundwater pumping, habitat loss, *Sporobolus airoides*, vegetation change, water table.

On July 6, 2007, lightning strikes along the base of the eastern escarpment of the Sierra Nevada in Owens Valley, California, started several fires. They became known as the Inyo Complex fire as they coalesced and burned >14,000 ha, almost destroying the towns of Independence and Big Pine. The fires started in sagebrush/bitterbrush shrubland high on alluvial fans and were blown by strong winds downslope through Mojave Desert shrubland to burn alkali meadow habitat at the fan toes.

Re-growth after the fire varied considerably in the alkali meadow. In some areas we observed native perennial $C_4$ grasses, *Sporobolus airoides* (alkali sacaton) and *Distichlis spicata* (saltgrass, nomenclature follows Baldwin et al. 2002), re-sprouting within days after the fire. In other areas we saw no re-growth. After several weeks, grass re-growth was so robust in some areas it was difficult to tell that a fire had occurred.

Alkali meadow habitat is rare in California (Sawyer and Keeler-Wolf 1995), and there are few quantitative data regarding its response to fire. Anecdotal observations made after other recent Owens Valley fires suggest *S. airoides* and *D. spicata* recover rapidly in areas of shallow groundwater, but we had not had the opportunity to closely monitor recovery after fires in areas subject to groundwater drawdowns. Understanding the response of alkali meadow to fire in areas of lowered water table is important because drawn-down water tables underlie much of this habitat in Owens Valley. The Inyo Complex fire burned a portion of alkali meadow habitat which covered a large gradient in depth-to-water table (DTW) at the time of the fire. It thus offered an unplanned opportunity to observe alkali meadow response to fire in areas of both shallow and deeply drawn-down groundwater.

Our primary objectives were to: 1) document species composition, cover, frequency, and phenology of the observed post-fire growth throughout the burned portion of the study area; 2) compare characteristics of pre- and post-fire vegetation; and 3) determine the extent to which DTW could account for vegetation patterns in the study area.

## METHODS

### Study Site

Owens Valley lies on the western edge of the Great Basin in eastern California at an elevation

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of approximately 1200 m above mean sea level. The Sierra Nevada borders the valley on the west, and the White and Inyo Mountains flank its eastern side. Owens Valley summers are hot and dry; winters are cold, and more than three-quarters of average annual precipitation falls during the October to March cold period. Although the climate is arid, with average annual precipitation of 130 mm, abundant water from Sierran snowmelt flows into Owens Valley and annually recharges shallow groundwater aquifers which underlie thousands of hectares of the valley floor (Hollett et al. 1991). As a result, valley floor vegetation is dominated by phreatophytic plant species assembled in alkali meadow and sink habitats (City of Los Angeles and County of Inyo 1991; Howald 2000). The growing season for alkali meadow species in Owens Valley commences in late March (first of spring), reaches a peak in late June when leaves and stems are fully grown, then enters winter dormancy in early October (Sorenson et al. 1991).

Alkali meadow vegetation is characterized as groundwater dependent because its estimated actual evapotranspiration exceeds annual precipitation (City of Los Angeles and County of Inyo 1991). Under the classification of groundwater-dependent ecosystems by Eamus et al. (2006), alkali meadow habitat falls under the "phreatophytic" category, because groundwater is not regularly expressed on the ground surface. In Owens Valley, large expanses of alkali meadow occur at the toes of Sierran alluvial fans located several miles from open water or Owens River, and the study site is in such a setting.

Most alkali meadow habitat in Owens Valley is on land owned by the City of Los Angeles and is managed for grazing and water export to Los Angeles by the Los Angeles Department of Water and Power. In the mid 1980's, "parcels" of relatively homogeneous vegetation in Los Angeles' holdings were delimited, mapped, and inventoried for species composition and cover. Groundwater levels and vegetation cover and composition on selected parcels have been regularly monitored since 1991. The study site (Fig. 1), located approximately 12 km north of the town of Independence, consists of two adjoining parcels classified in the 1980's as alkali meadow, a classification consistent with Lee's (1912) "grassland" classification made over 70 yr before. Both study site parcels were dominated before the fire by two native phreatophytic grasses *S. airoides* and *D. spicata*, and, in places, by the encroaching shrubs *Artemisia tridentata*, *Atriplex lentiformis* subsp. *torreyi*, and/or *Chrysothamnus nauseosus*. Historically, livestock grazing had occurred throughout the study area (typically from October through May), including during spring 2007. After the July 2007 fire, livestock were excluded from the study area for the remainder of the 2007 growing season.

Substrate of the study site is a combination of ancient beach, bar, or river channel sediments. These deposits lie between alluvial fan deposits upslope to the west and fluvio/lacustrine deposits of the valley floor downslope to the east (Hollett et al. 1991). Soils are moderately- to poorly-drained loams (USDA Natural Resources Conservation Service 2002).

## Vegetation Sampling

Inyo County Water Department (ICWD) monitors vegetation composition and cover each summer along randomly-located 50-m transects in selected parcels. ICWD uses the point-intercept technique (Bonham 1989) to determine species composition of the top canopy layer at 0.5-m intervals along the transect. The endpoints of these transects are not permanently marked, but the starting location is recorded with a GPS unit, and its randomly-generated compass bearing is recorded. Two weeks before the Inyo Complex fire, ICWD read point-intercept transects at 48 random locations throughout the two parcels comprising the study area (Fig. 1).

We conducted visual reconnaissance of the burned area several times, starting one week after the fire. Eight weeks after the fire, August 30–September 6, 2007, we used GPS and compass bearings to approximately relocate then re-sample the 48 pre-fire random transect locations in the study area. Twenty were in burned areas and 24 in un-burnt areas. Four of the 48 transects were not sampled because they crossed the burn boundary. To increase the size and spatial extent of post-fire sampling, we randomly selected 22 additional transects throughout the study site. Altogether, we obtained data from 37 transects in the burned area and 29 from the unburned "control" area after the fire (Fig. 1). We employed the same point-intercept method used before the fire to measure species composition and estimate top layer canopy cover. We also noted the most advanced phenology of each species sampled along each transect in the burned area. Phenology was not recorded along pre-fire transects nor along transects in the unburned control area.

## Precipitation and Water Table Data

ICWD maintains a precipitation gauge in the study area, and its precipitation totaled 31 mm for the period October 1, 2006–April 30, 2007. No precipitation was recorded from April 17, 2007, until August 28–30, when 3 mm were recorded. Average precipitation for the ICWD gauge period of record (1993–2006) was 160 mm.

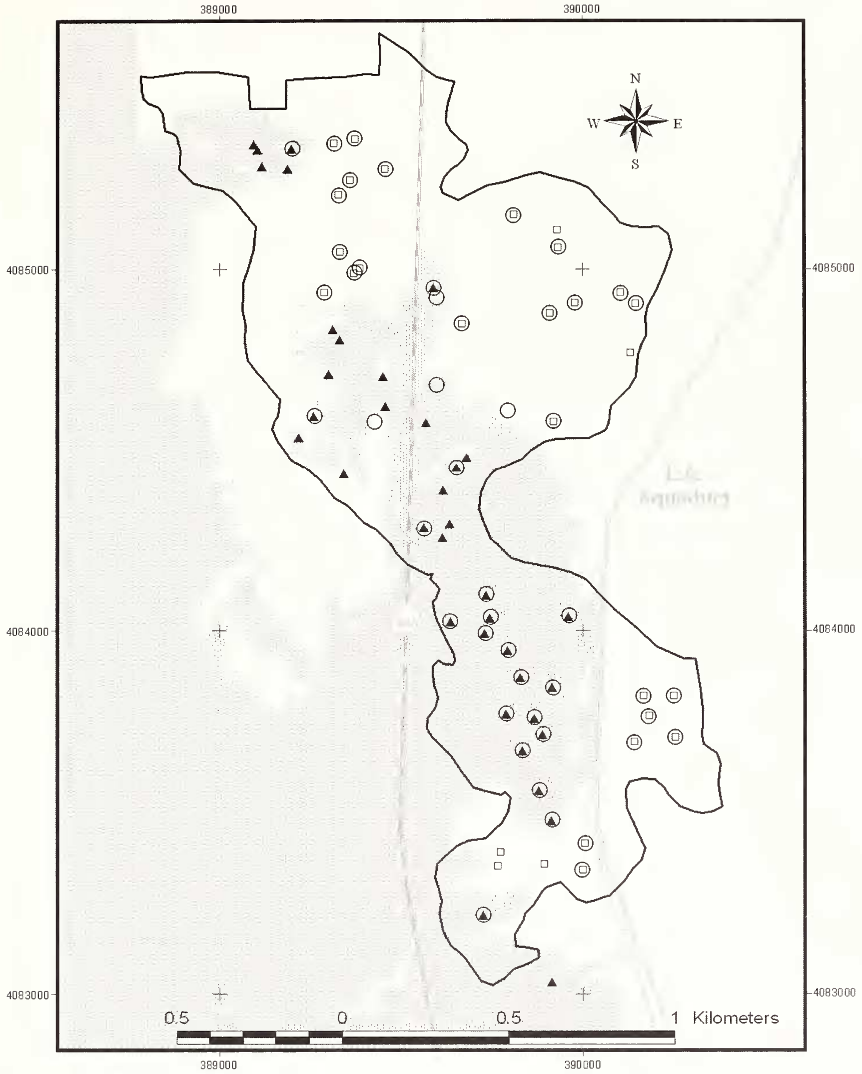

FIG. 1. Study area and sample sites. The 204-ha study area is outlined in black. The burned portion is indicated by stippling and covers 76 ha. Open circles represent start points of transects read before the fire; solid triangles represent start points of transects read after the fire in the burned area; open squares represent start points of transects in the unburned (control) area. Map projection: UTM Zone 11, NAD 27.

To show the historic DTW gradient in the study area we digitized the 8 ft (2.4 m) and 4 ft (1.2 m) contours from Lee's (1912) map and included them in Fig. 2. Lee (1912) estimated DTW in southern Owens Valley during the period 1908–1911 based on readings of 169 observation wells, seven of which were in or adjacent to the study area. Lee gathered his data before large scale groundwater pumping had been initiated in Owens Valley.

To show the 2007 DTW gradient, we obtained DTW data collected on or near April 1, 2007, from 14 observation wells in or immediately adjacent to the study area (Fig. 2). We chose April 1 DTW data because early spring groundwater levels typically represent the annual high stand before decline due to the onset of seasonal evapotranspiration (Lee 1912), and because they are used by ICWD and other researchers for tracking annual changes in groundwater depth as well as for investigations of DTW-cover relationships (Elmore et al. 2006). Next, we applied ordinary kriging, as implemented in ArcGIS 9.2, Spatial Analyst extension software (circular variogram, 10-m grid cells, and default values for other parameters), to the April 2007 DTW data to create a DTW grid covering the study area. We chose kriging as opposed to other interpolation techniques because it is recommended in cases where sample sites are irregularly spaced (Legendre and Legendre 1998). We derived DTW

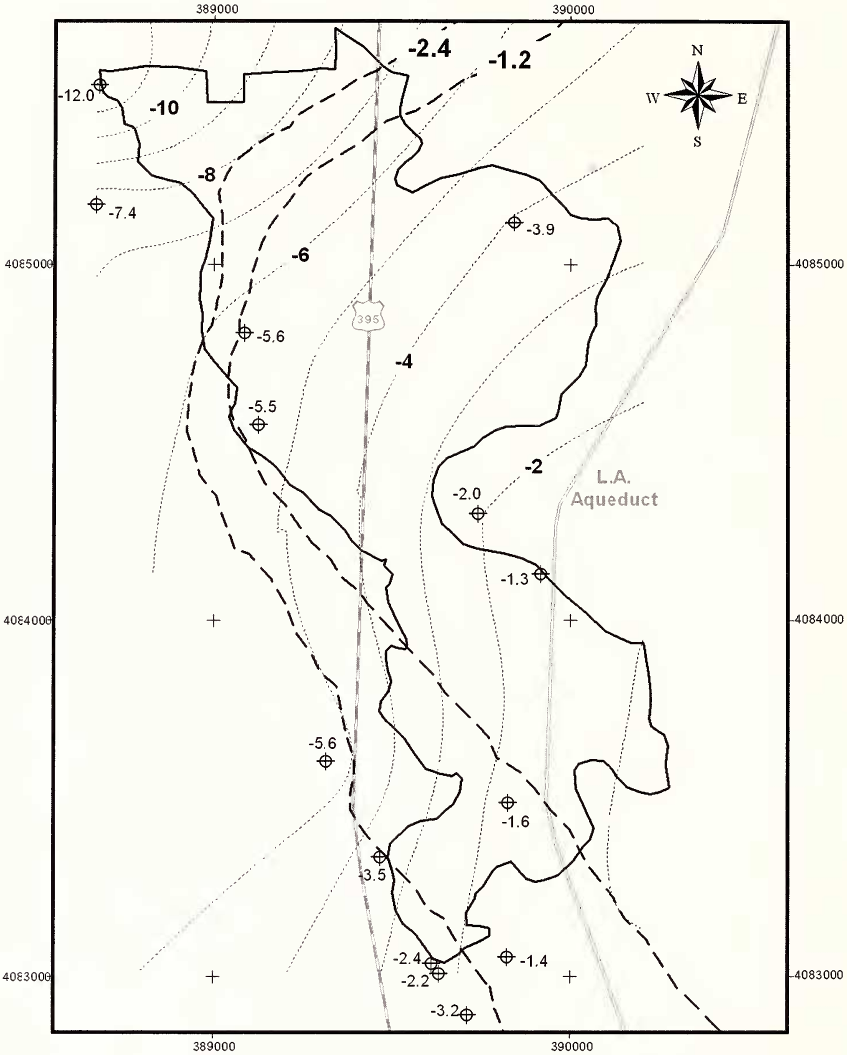

FIG. 2. Historic and 2007 water table gradient. Solid black line outlines study area. Dotted lines represent April 2007 water table contours (labeled in meters from ground surface). Bold-faced dashed lines represent the −8 ft (−2.4 m) and −4 ft (−1.2 m) water table contours mapped by Lee (1912). Crossed circles represent monitoring wells used for kriging, and adjacent numbers represent April 2007 DTW values at the wells. Scale and map projection as in Fig. 1.

contours from the grid and included them in Fig. 2.

The current DTW gradient in the study area—with drawdowns of up to 5 m and alterations of subsurface flow relative to conditions mapped by Lee (1912)—is controlled by large scale groundwater extraction which began in 1970 (Fig. 3) (City of Los Angeles and County of Inyo 1991; Steinwand and Harrington 2003a, b; City of Los Angeles Department of Water and Power and Inyo County 2007).

To estimate DTW under each vegetation transect, we overlaid the 2007 transects on the kriged 2007 DTW grid and calculated the DTW as the mean of all grid cells intersected by the transect.

## Analysis

We grouped transect data into three sets for analysis: 1) pre-fire data covering the entire study area; 2) post-fire data from the burned portion of the study area; and 3) post-fire data from the unburned control portion of the study area. To characterize vegetation we calculated each species' mean cover and frequency based on the transects sampled in each dataset. We used a bootstrap technique to estimate standard errors because of the skewed distributions of most species (Elzinga et al. 1998). We calculated the Mann-Whitney U statistic to determine if the mean cover of *S. airoides*, the dominant species in the post-fire burned area, differed from its cover

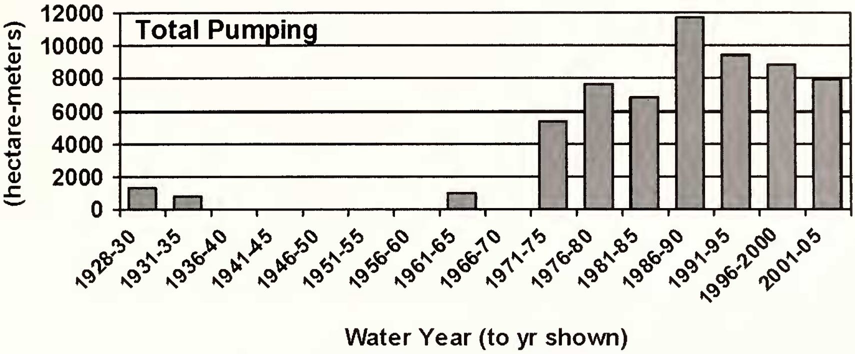

FIG. 3. Groundwater pumping from wellfield in which study site is located. Amounts (in ha-m) are shown as five water-year cumulative totals, 1931–2005.

in the pre-fire and post-fire control datasets. We calculated the chi square statistic to find out if frequencies of the three most common species in the post-fire burned area differed significantly from their frequencies in the pre-fire and post-fire control datasets. We calculated the Mann-Whitney U statistic to determine if there was a significant difference in mean DTW between post-fire burned transects with flowering *S. airoides* versus those with vegetative-only individuals. To allow visual comparison of cover and frequency of each species among the three datasets we juxtaposed data in a table.

We used ordinary least-squares regression (as implemented in Sigma Plot 10.0 software) to attempt to explain variance in grass cover (summed cover of *S. airoides* and *D. spicata* at each transect) as a function of estimated April 2007 DTW at each transect in each dataset. Each transect's grass cover and DTW represented a single data point in these regression analyses. We performed exploratory linear regressions on raw data, then transformed grass cover data by adding one to each value and taking the natural log of the sum. We then regressed the transformed data against DTW using a sigmoid curve. We chose the sigmoid curve rather than the straight line for both practical and theoretical reasons: it explained more variance, and species abundance can be better related to environmental gradients by unimodal curves (ter Braak 1996), of which sigmoid curves are special cases (ter Braak and Looman 1995).

## RESULTS

### Fire Effects

At our first site reconnaissance within days after the fire, virtually all above-ground biomass had been reduced to ash or charred wood. By eight weeks after the fire, seven species showed measurable re-growth in the burned area (Table 1). Total transect cover ranged from zero to 38%. *Sporobolius airoides* and *D. spicata* were the dominant graminoid species, dominant overall, and showed great spatial variation in cover. *Glycyrrhiza lepidota* was the third most abundant herb. *Sporobolius airoides* and *Heliotropium curassavicum* sampled in burned-area post-fire transects were in both vegetative and flowering states, while we sampled *D. spicata* only in vegetative state.

Average cover of *S. airoides*, the dominant species in the burned area after the fire, was significantly lower than *S. airoides* measured in pre-fire and post-fire control datasets ($P < 0.05$) (Table 1). However, post-fire frequencies of the three most common species in the burned area did not differ significantly from frequencies in the pre-fire and post-fire un-burned control area datasets (Table 1). Although most graminoid species recorded before the fire were recorded after the fire, few other herbaceous and woody species were sampled on post-fire transects in the burned area. A single hit on *Salix exigua* was the only documentation of post-fire shrub re-growth (Table 1).

### DTW Effects

Grass cover was significantly correlated with DTW in all three datatsets ($P < 0.01$). DTW explained 87% of the variance in pre-fire grass cover, 77% of the variance in the post-fire burned area, and 94% of the variance in the post-fire control area (Fig. 4).

The mean DTW for transects with flowering *S. airoides* was 2.6 m (SE = 0.20) and was shallower than the mean for transects in which *S. airoides* was in vegetative state only, 3.6 m (SE = 0.32), and the difference was significant ($P < 0.05$). *Heliotropium curassavicum* was also observed flowering along transects after the fire, but the sample size was too low for statistical analysis.

TABLE 1. SPECIES COMPOSITION, COVER, AND FREQUENCY IN PRE- AND POST-FIRE VEGETATION SURVEYS. (See Fig. 1 for sampling locations). Growth forms are: graminoid (g), other herbaceous (h), and shrub (s). "Num. trans." indicates number of transects on which a given species was hit. "Trans. freq." was calculated by dividing the total transects by "Num. trans.". "Max. cov." is the highest cover of a given species on a single transect. Asterisk (*) designates post-fire cover value significantly lower than pre-fire and post-fire control values as determined by Mann-Whitney U test ($P < 0.05$).

| | | Pre-fire entire site, June 2007 48 transects | | | | | Post-fire burned, August 2007 37 transects | | | | | Post-fire unburned, August 2007 29 transects | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Form | Plant species | Pct. cov. | Std. err. | Num. trans. | Trans. freq. | Max. cov. | Pct. cov. | Std. err. | Num. trans. | Trans. freq. | Max. cov. | Pct. cov. | Std. err. | Num. trans. | Trans. freq. | Max. cov. |
| g | *Sporobolus airoides* | 16.7 | 2.2 | 37 | 0.77 | 45 | 6.1* | 1.2 | 25 | 0.66 | 32 | 13.5 | 2.5 | 20 | 0.69 | 49 |
| g | *Distichlis spicata* | 6.6 | 1.0 | 36 | 0.75 | 27 | 3.2 | 0.6 | 24 | 0.63 | 15 | 5.6 | 1.7 | 19 | 0.66 | 34 |
| g | *Juncus balticus* | 2.2 | 0.6 | 17 | 0.35 | 15 | <0.1 | | 1 | 0.03 | | 0.9 | | 7 | 0.24 | 9 |
| g | *Carex* sp. | <0.1 | | 2 | 0.04 | | 0.0 | | | | | 0.0 | | | | |
| h | *Glycyrrhiza lepidota* | 2.4 | 0.9 | 14 | 0.29 | 34 | 1.2 | 0.5 | 9 | 0.24 | 13 | 0.8 | | 9 | 0.31 | 6 |
| h | *Bassia hyssopifolia* | 0.1 | | 2 | 0.04 | | 0.0 | | | | | 0.2 | | 1 | 0.03 | |
| h | *Heliotropium curassavicum* | 0.2 | | 2 | 0.04 | | 0.3 | | 3 | 0.08 | | 0.0 | | | | |
| h | *Malvella leprosa* | <0.1 | | 2 | 0.04 | | 0.0 | | | | | 0.0 | | | | |
| h | *Nitrophila occidentalis* | <0.1 | | 2 | 0.04 | | 0.0 | | | | | 0.0 | | | | |
| h | *Asclepias fascicularis* | <0.1 | | 1 | 0.02 | | <0.1 | | 1 | 0.03 | | 0.0 | | | | |
| h | *Iva axillaris* | <0.1 | | 1 | 0.02 | | 0.0 | | | | | 0.0 | | | | |
| h | *Pyrrocoma racemosa* | <0.1 | | 1 | 0.02 | | 0.0 | | | | | <0.1 | | 1 | 0.03 | |
| sub-s | *Machaeranthera carnosa* | 0.4 | | 5 | 0.10 | | 0.0 | | | | | 1.5 | | 8 | 0.28 | |
| s | *Chrysothamnus nauseosus* | 2.7 | 0.7 | 25 | 0.52 | | 0.0 | | | | | 1.7 | 0.4 | 16 | 0.55 | 5 |
| s | *Atriplex lentiformis* subsp. *torreyi* | 2.5 | 0.8 | 21 | 0.44 | | 0.0 | | | | | 1.0 | 0.3 | 11 | 0.38 | 8 |
| s | *Artemisia tridentata* subsp. *tridentata* | 2.7 | 0.8 | 17 | 0.35 | | 0.0 | | | | | 3.8 | 0.9 | 13 | 0.45 | 17 |
| s | *Stephanomeria* sp. | <0.1 | | 2 | 0.04 | | 0.0 | | | | | 0.2 | | 4 | 0.12 | |
| s | *Atriplex canescens* | <0.1 | | 1 | 0.02 | | 0.0 | | | | | 0.0 | | | | |
| s | *Psorothamnus polydenius* | <0.1 | | 1 | 0.02 | | 0.0 | | | | | 0.0 | | | | |
| s | *Salix exigua* | 0.2 | | 1 | 0.02 | | <0.1 | | 1 | 0.03 | | 0.1 | | 1 | 0.03 | |
| s | *Sarcobatus vermiculatus* | <0.1 | | 1 | 0.02 | | 0.0 | | | | | 0.2 | | 2 | 0.07 | |
| s | *Suaeda moquinii* | 0.1 | | 1 | 0.02 | | 0.0 | | | | | 0.2 | | 3 | 0.10 | |

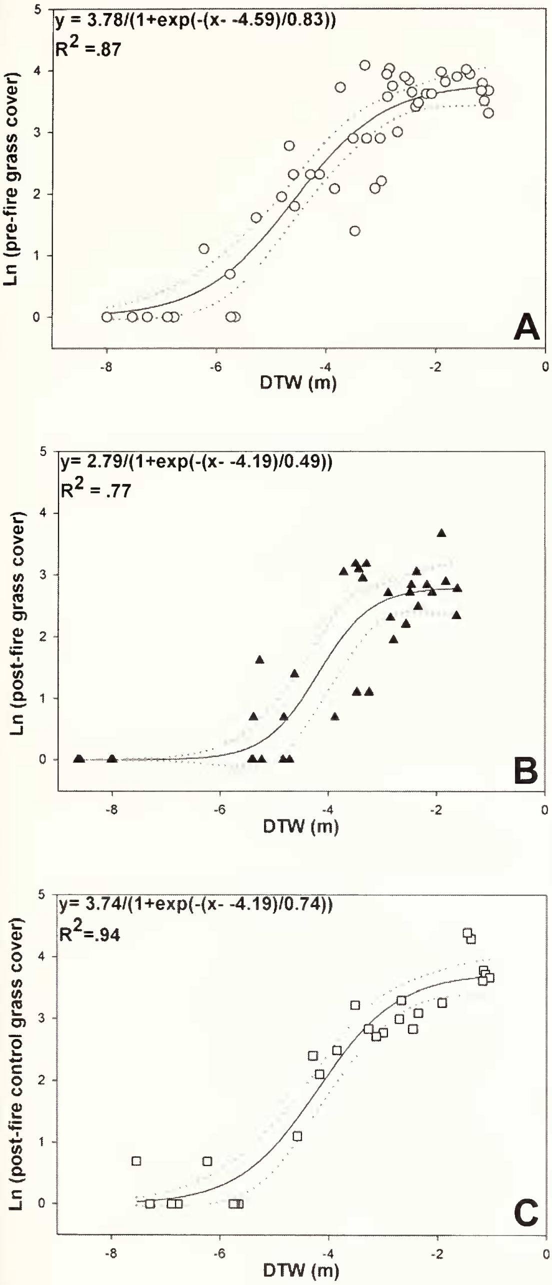

FIG. 4. Grass cover as a function of DTW in meters below ground surface. Dotted lines represent 95% confidence limits: (A) Pre-fire dataset, n = 48; (B) Post-fire burned area dataset, n = 37; (C) Post-fire unburned control dataset, n = 29. All coefficients were significant at $P < 0.01$.

## DISCUSSION

### Fire Effects

The fire's timing in summer, the noted thoroughness of combustion of above-ground biomass, and the meager precipitation in 2007 (both before the fire and in the eight weeks after the fire), led us to expect little re-growth before the 2008 growing season (following winter precipitation). We sampled vegetation just eight weeks after the fire because, contrary to our expectations, we had observed vigorous re-growth in places, and we desired to document the species showing re-growth and investigate the spatial variation in that re-growth. We had no expectation that re-growth was complete and, therefore, expected mean cover over the burned area to be lower than mean cover of the pre-burn and post-burn control areas (Table 1). A noteworthy finding was that short-term re-growth of two grasses (and the herb *G. lepidota*) did attain pre-fire and post-fire control levels in terms of frequency (Table 1), suggesting the fire caused little mortality among the dominant herbaceous species. This finding is consistent with the results of Smith and Kadlec (1985), who found no substantial below-ground mortality of saltgrass after burning in a Utah marsh.

The four graminoid and forb species which attained greater than trace cover values after the fire in the burned area (Table 1) show anatomical/physiological traits which facilitate survival after fire. Perennating buds occur below ground on rhizomes for *D. spicata* and at the root crown for the bunchgrass, *S. airoides*. *Glycyrrhiza lepidota* is rhizomatous while *H. curassavicum* has underground rootstocks able to give rise to new stems. For these reasons, in addition to the dry, hot, post-fire conditions which made seed germination and seedling survival extremely unlikely, we treated live cover sampled in the post-fire burned area as re-growth from plants already established before the fire, rather than new recruits.

In contrast to the re-growth of herbaceous species, shrubs may have suffered high mortality. *Artemisia tridentata* is known to be killed by fire and re-establish by seed (Tirmenstein 1999a). The absence of *C. nauseosus* after the fire (Table 1) suggests mortality because it has been reported by several researchers to show "rapid" post-fire recovery through re-sprouting as well as by seed germination (Tirmenstein 1999b), and we have observed it re-sprouting within weeks after other Owens Valley fires. Fire effects on *A. lentiformis* subsp. *torreyi* have not been systematically studied, though some data suggest post-fire recovery is by seed germination rather than re-sprouting (Meyer 2005). Observations in Owens Valley show high rates of *A. lentiformis* subsp. *torreyi* germination in spring following winters with abundant precipitation (ICWD data on file). We found no data regarding fire effects on *Machaeranthera carnosa*. Overall, the absence of most shrub species in our observations and sampling data eight weeks after the fire suggests shrub mortality.

### DTW Effects

One of the principal reasons we sampled post-fire vegetation was to investigate the spatial variation in re-growth observed in reconnaissance site visits. DTW explained most of the variance (Fig. 4b) in grass cover in our post-fire sample of the burned area. The significant difference in DTW between flowering and vegetative *S. airoides* samples is further evidence for the importance of DTW—as opposed to properties of the fire itself—in controlling patterns of re-growth. The fact that DTW also explained most of the variance in grass cover in both pre-fire and post-fire control datasets (Figs. 4a, 4c) suggests the post-fire recovery in the burned area is re-establishing cover—DTW relationships which existed before the fire. These patterns had previously been obscured from view by encroaching shrubs.

These relationships between grass cover, phenology, and DTW may be understood as effects of differential water availability across the DTW gradient in both burned and un-burned portions of the study area. Cox et al. (1990) asserted that a "waving sea" of alkali sacaton could not be maintained where annual precipitation ranges from 150–400 mm, while revegetation experiments elsewhere in Owens Valley have documented water requirements for *D. spicata* to range from 200–800 mm/yr (Dickey et al. 2005). However, with average annual precipitation of 160 mm and only 34 mm total precipitation prior to and during the 2007 growing season, a beautiful, "waving sea" of alkali sacaton and saltgrass developed in the burned area at the shallow end of the DTW gradient eight weeks after the fire, but not at the drawn-down end. The simplest explanation for the observed patterns is differential groundwater availability.

Livestock grazing is another important factor potentially controlling patterns of cover and phenology, but because livestock were not permitted in the study area after the fire, low cover in parts of the burned area was not a direct result of grazing. The pattern of un-grazed post-fire grass re-growth was similar to the pattern of grazed pre-fire grass distribution. While grazing undoubtedly affects the site, our results argue against differential grazing as a direct explanation for the observed variation in grass cover.

We are not the first to infer the importance of DTW as a primary factor in controlling vegetation patterns in arid lands. Los Angeles Department of Water and Power engineer C. H. Lee (1912) documented the "very striking" relationship between vegetation and DTW in this portion of Owens Valley nearly a century ago, and groundwater availability is used to explain vegetation patterns on valley floors throughout the Great Basin (Coville 1893; Odion et al. 1992; Castelli et al. 2000; West and Young 2000; Cooper et al. 2006).

The 2007 DTW gradient, which explains a high percentage of current variation in grass cover, is controlled by groundwater extraction (previously noted) and is quite different from the gradient Lee mapped in 1912 (Fig 2). The 2007 distribution in grass cover, therefore, is best interpreted as a response to this altered hydrologic regime. Elmore et al. (2006) explained changes in perennial cover of Owens Valley alkali meadow habitat over 20 yr as a function of changes in DTW and precipitation. Cooper et al. (2006) described a similar case in Colorado. In our study we substituted space for time by examining grass cover before and after fire in a single growing season and obtained similar results.

### Management Implications

At the shallow end of the DTW gradient (Fig. 2) the robust recovery of native grass provided dramatic evidence that fire can serve as a force for meadow regeneration. This finding is not surprising: burning to remove shrubs and promote grasses has been practiced in California since pre Euro-American settlement (Anderson 2006).

More relevant to our research, Wright and Chambers (2002) conducted and studied results of controlled burns in Great Basin riparian meadow habitat located in both shallow and drawn-down water table sites. They concluded that burning is an effective tool for restoring meadows with shallow groundwater.

At the drawn-down end of the DTW gradient, fire may be an agent of community change rather than a force for meadow regeneration. We expect vegetation there will cease to meet criteria for classification as groundwater dependent meadow and, instead, will come to resemble that of nearby precipitation-dependent desert shrublands. The expected multi-year period for shrub re-colonization, combined with negligible grass cover, will leave the drawn-down end of the DTW gradient highly vulnerable to erosion. In Owens Valley, wind poses an especially great erosional force, and a major highway (US 395) was closed on at least one occasion after the fire due to the density of airborne meadow soil where the highway crossed the study area. Other studies predict vegetation change as a response to hydrologic alterations under phreatophytic communities (Odion et al. 1992; Manning 1999; Wright and Chambers 2002; Naumburg et al. 2005; Patten et al. 2008). Our data suggest fire may accelerate the rate of the predicted vegetation change.

### Conclusion

We conclude that if the 2007 DTW gradient is maintained in the future, vegetation at opposite ends of the gradient will follow different trajectories. At the shallow end we expect grass cover will reach or exceed pre-fire levels in summer 2008 (preliminary data suggest this outcome) and alkali meadow habitat will be maintained. At the drawn-down end the fire will facilitate the conversion of groundwater-dependent alkali meadow to a more xeric community of annuals and, eventually, shrubs. Rather than acting as an independent force shaping patterns of vegetation, our short-term response data suggest the effect of the fire over the entire DTW gradient will be the long-term amplification of differences already present before the fire. These differences are associated with differential water availability. Our examination of fire effects led to documentation of groundwater dependence and the effects of groundwater limitation.

## ACKNOWLEDGMENTS

We thank Jerry Zatorski and Sondra Grimm of the Inyo County Water Department for conducting the field work. We also thank the University of California White Mountain Research Station, John Callaway, and an anonymous reviewer.

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